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Integrating human health impact from indoor emissions into an LCA: a case study evaluating the significance of the use stage

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Abstract
Purpose Indoor emissions of toxic substances from products can have a negative effect on human health. These are typically not considered in a life cycle assessment (LCA), potentially underestimating the importance of the use stage. The purpose of this paper is to develop a method that, based on a set of measured emission rates, calculates the impact on human health during the use stage of products that are used indoors and that emit volatile organic compounds (VOCs).
Methods Emissions from a product are measured in a test chamber and reported as a set of emission rates (microgrammes per hour) at specific points in time (hour/day). Constrained non-linear regression (CNLR) analysis is then used to determine parameters for three emission models, and a model is selected based on goodness of fit with the measured emission rates (R^2 and expert judgement). The emission model is integrated over a defined time period to estimate the total use stage emissions per functional unit (FU). The total emissions are subsequently integrated in a homogeneously mixed one-box model within the USEtox model. Intake fraction (iF) is calculated based on size of residential home, inhalation rate, exposure time, ventilation rate, mixing factor and number of people exposed.
Results and discussion The method is tested in a case study of a chair, with the results showing that the impacts in the use stage are in most cases significantly higher than from the production and disposal stages combined. The sensitivity to parameter variations is evaluated. Intake fraction (factor of 761), replacement frequency (factor of 70) and emission model (factor of 24) are found to be the most important model parameters. Limiting early exposure (>14 % of emissions may occur in the first month and >50 % in the first year) and replacing furniture less frequently will reduce exposure.
Conclusions The case study shows that the impact on human health from indoor emissions can be of significance, when compared to the impact on human health from total outdoor emissions. Without specific exposure data (e.g. ventilation rates) the uncertainty will be high. The developed method is applicable to all products that emit VOCs, provided that the emission rate can be modelled using an exponential decay model and that the product amount is related to a meaningful functional unit. It is recommended that when performing an LCA of products that emit VOCs, the indoor use stage is included in the life cycle impact assessment.

Keywords Furniture · Human health · Indoor air quality · Indoor emission modelling · Life cycle impact assessment · USEtox

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1 Introduction

Indoor emissions of toxic substances can have a significantly higher impact on human health than outdoor emissions. This is due in part to indoor concentrations being higher than outdoor concentrations (Sexton et al. 2003) and in part to people spending the majority of their time indoors

(Brasche and Bischof 2005). This hazard is well-known; reducing indoor emissions of toxic substances such as formaldehyde from building materials has been a goal for more than four decades (Salthammer et al. 2010). Although these reduction efforts have been successful, indoor concentrations in residential homes have been reduced to a lesser degree than anticipated (Hun et al. 2010). This is an indication of additional emission sources being present. This is supported by studies showing that furniture can be a significant source of formaldehyde (Blondel and Plaisance 2011).

Public demand for stronger regulation of emissions from building materials and products (e.g. furniture, carpets) is continually growing (Kuehn 2008), and labelling schemes have also addressed this issue (Nordic Ecolabelling 2011a, b; Greenguard 2011). However, in most instances, these are based on risk assessments that use emission rates measured within a specific number of days after production, rather than over the life of the product. An example of this is the Nordic Swan requirements for building, decoration and furniture panels, where formaldehyde emissions must be below 0.065 mg/m^3 (Nordic Ecolabelling 2011b) when measured after 28 days with the M1 testing protocol (Saarela and Tirkkonen 2004). The long-term effect of low-level exposure is typically not assessed, and the results are not related to impacts of other stages in the life cycle. This may lead to problem shifting from environmental to human health aspects or from one life cycle stage to another. Including these impacts in a life cycle assessment (LCA) will influence the final result, and 'could even lead to human toxicity becoming a dominant impact category for certain products such as paints, furniture, or carpets' (Hellweg et al. 2009).

The goal of this paper is to develop a method that, based on a set of measured emission rates, can be used to calculate the impacts on human health during a product's use stage. This will be done by first determining the most significant parameters for modelling emissions, exposure and human health impact, respectively. Secondly, the significance of the use stage will be evaluated empirically in a case study of a chair. This work builds on previous research on indoor emission modelling (Guo 2002a, b), Hellweg et al.'s (2009) proposed framework for integrating indoor exposure with life cycle impact assessment (LCIA) and the UNEP/SETAC toxicity model (Rosenbaum et al. 2008). The model is intended to be applied to products that are used indoors and that emit volatile organic compounds (VOCs). Examples of such products can be furniture, toys, paints and carpets.

2 Methods

Including the impact on human health from indoor emissions into the LCIA phase of an LCA requires that an emissions inventory is established, and that for each substance in the inventory a characterisation factor is developed (as shown in Fig. 1). For this purpose, a combination of existing models from occupational health, toxicology and life cycle assessment is used. The steps needed to go from emission rates to impact on human health are described below.

2.1 Emission rates and emission models

Emissions from a product are measured in a test chamber and reported as a set of emission rates (microgrammes per hour) at specific points in time (hour/day). This is in contrast to the LCA methodology where the focus is on the total input and output, not on rates (International Organization for Standardization 2006a). The total output is therefore found by integrating a time-dependent emission model over a defined time period.

Selecting an emission model that fits is limited by our lack of knowledge of the diffusivity of the product, of the initial amount of VOC in the product and of any characteristics of the room it will end up in (volume, air exchange, and loading). Guo (2002a) reviewed 52 indoor emissions source models. This comprehensive review is here the basis for identifying applicable emission models for calculating a product's use stage emissions. The majority of the models are intended for specific applications (e.g. emissions from surface coating), for instant or constant emissions, or are pure data-fitting models. These are considered not applicable for general emissions modelling. Of the six remaining models, three are not included because they are equivalent to one of the other models and thus redundant. The three chosen models are all decay models based on emission rate being controlled by internal diffusion, and not limited by vapour pressure. This assumption is considered reasonable as we are interested in total mass emitted, and not in concentrations (Tichenor and Guo 1991). The models are presented below. Equation (1) is a first-order decay model (Guo 2002a), Eq. (2) is an *n*th-order decay model (Tichenor et al.

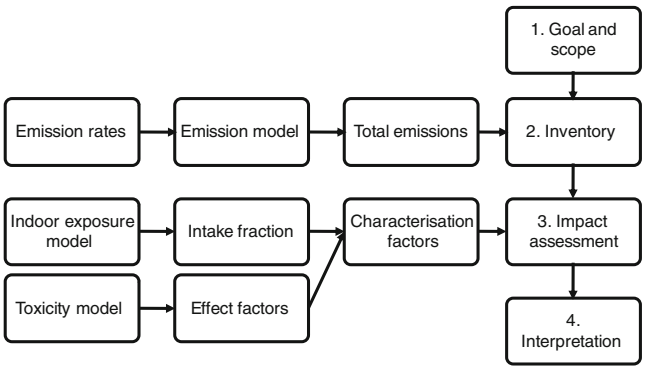

Fig. 1 Integrating indoor emissions into LCA

1992) and Eq. (3) is a dual first-order decay model (Guo 2002a).

$$E = E_0 \cdot e^{-kt} \tag{1}$$

$$E = \frac{E_0}{\left[\left(1 + (n-1) \cdot k \cdot t \cdot E_0^{n-1}\right)^{\frac{1}{(n-1)}}\right]} \tag{2}$$

$$E = E_1 e^{-k_1 t} + E_2 e^{-k_2 t} \tag{3}$$

where E is emission rate and E_0, E_1 and E_2 are initial emission rates, k, k_1 and k_2 are decay rates and t is time. Unknown parameters are found using constrained non-linear regression analysis (CNLR) (SPSS 2010). This requires that a minimum of three to five data points are measured, as there are two to four unknown parameters. A specific indoor emission model can then be selected, based on an evaluation of the goodness of fit. Note that the indoor emission models originally refer to emission factor (e.g. micrograms per square metre per time) and not emission rate per product (micrograms per time), but the emission factor and emission rate are proportional.

2.2 Indoor exposure model

Following the recommendations of Hellweg et al. (2009), a homogeneously mixed one-box model is used to model indoor exposure from a product. The model is connected to the surroundings through ventilation. The model is further simplified by disregarding any concentrations of substances in the ingoing ventilation air, limiting the focus to the additional impact of the product.

Inhalation is assumed to be the most significant exposure pathway, thus excluding dermal contact and ingestion from the assessment (Meijer et al. 2005). To estimate the impact on human health, we must know how much of the total emitted mass of pollutants is inhaled by human beings. This ratio is defined as the intake fraction (iF) (Bennett et al. 2002). The intake fraction (dimensionless) is calculated using Eq. (4), based on Hellweg et al. (2009):

$$iF = \frac{IR}{V \cdot k_m \cdot k_{ex}} \cdot N = \frac{IR \cdot h}{V \cdot k_m \cdot 24 \cdot k_{ex}} \tag{4}$$

where IR is inhalation rate (cubic metre per hour), h is exposure time (hour/day), V is room volume (cubic metre), k_m is the mixing factor (how well the air is mixed in the room) (dimensionless), 24 is hours per day (hour/day), k_{ex} is the air exchange rate per hour (1/h) and N is the number of people exposed. Assuming one person per product eliminates N from the equation. The iF is used to calculate the inhaled mass, using Eq. (5):

$$m_{inh} = m_{tot} \cdot iF \tag{5}$$

where m_{inh} is the inhaled fraction of the total output, m_{tot}. The total output is, as previously mentioned, calculated by integrating a time-dependent emission model over a defined time period. When performing an LCA, the calculation of m_{tot} should be related to the functional unit (FU) and its reference flows (e.g. are we interested in the total emissions from one product over 10 years or the total emissions from two products over 5 years each?). In this case, m_{tot} becomes mass per FU.

Adsorption and desorption of vapour phase organic compounds influence the concentration variations in real rooms (Singer et al. 2007). In this case, the emission of VOC and aldehydes from the product will be able to adsorp on other surfaces in the real room, with subsequent desorption to the indoor air, when the room concentration decrease. This is well-proved (Tichenor et al. 1991; Colombo et al. 1993; Jørgensen et al. 1999; Jørgensen and Bjørseth 1999). If the product is complex, a combined sink-diffusion model could be even better model than a simple Langmuir Isotherm model (Jørgensen et al. 2000). Another aspect of sorption is that the product itself can be able to adsorp vapour phase organic compounds from other sources in the real room due to its own surface, with corresponding desorption when the room concentration decrease. This is also relevant for complex products, for example products with a large surface.

The influence of sorption on the indoor concentration depends on the sorption capacity of the chemical compounds involved and the material surfaces in the room. No standard method or standard values currently exists for inclusion of sorption parameters to the emission model for VOCs (Wenger et al. 2012). In general, more advanced models often fit the data better than simple models, but without knowledge of sorption data, it makes no sense to include sorption into the model used.

2.3 Outdoor exposure model

For outdoor exposure, the USEtox model with a nested indoor compartment is used (Hellweg et al. 2009). USEtox™ is a multimedia and multi-compartment environmental fate, exposure and effect model developed by UNEP/SETAC, based on scientific consensus (Rosenbaum et al. 2008). The indoor compartment is, as previously mentioned, linked with the outdoor model through air exchange. The mass of pollutants entering the environment is calculated as the total emitted mass minus that which is inhaled, as shown in Eq. (6).

$$m_{ua} = m_{tot} - m_{inh} = m_{tot} \cdot (1 - iF) \tag{6}$$

where m_{ua} is emissions to urban air per FU, with urban air having been selected as the receiving compartment in order to

simplify the model, a choice that is assumed to have a negligible effect on the total impact.

2.4 Use stage impact assessment

For every emission the potential indoor and outdoor impact is calculated using Eqs. (7) and (8):

$$Impact_{h,\text{indoor}} = m_{tot} \cdot iF \cdot EF_{inh} = m_{tot} \cdot CF_{indoor} \quad (7)$$

$$Impact_{h,\text{outdoor}} = m_{ua} \cdot CF_{ua} \quad (8)$$

where $\text{Impact}_{\text{h,indoor}}$ and $\text{Impact}_{\text{h,outdoor}}$ are the impacts on human health [cases per FU], EF is the human health effect factor for inhalation of the specific substance [cases per kilogramme intake] (cancer and non-cancer, see the USEtox model for more information (USEtox 2011)) and the characterisation factors (CFs) are the indoor and outdoor characterisation factors of the substances [cases per kilogramme emitted] (also termed comparative toxic units) (Rosenbaum et al. 2008).

The CF takes into account the potential fate, exposure and effect of the emissions. For indoor emissions, the characterisation factor is defined as intake fraction (iF) multiplied with effect factor (EF) (Hellweg et al. 2009). The intake fraction is the same for all emissions, but the effect factor is determined individually for every emitted substance based on the ED50 [kilogramme per lifetime] (Rosenbaum et al. 2008). For outdoor emissions, the characterisation factors are taken directly from the USEtox model.

3 Application of the method in a furniture case study

In order to evaluate the significance of impacts from emissions in the use stage, a case study of a chair was performed. The chair is in ordinary production, and can be described as a recliner. The base is made from steel and laminated European beech. The seat is made of upholstery leather, with the inside consisting mainly of polyurethane foam. The total weight of the chair (excluding packaging) is slightly more than 20 kg. First, the emissions from the chair were measured in a test chamber over a period of 28 days using standardised test methods (International Organization for Standardization 2001, 2004, 2006b, c), as described below. Based on these results the impacts on human health were calculated for 1, 3, 5, 15, 30 and 70 years using the methodology presented above (Eqs. (1), (2), (3), (4), (5), (6), (7) and (8)), with 8 h of indoor exposure time in residential homes per day. Eight hours is based on the assumption that a person is exposed to the emissions from the chair half of the approximately 16 h spent indoor at home (Brasche and Bischof 2005). Secondly, an LCA was performed according to the Product Category Rules (PCR) for chairs (EPD Norway 2008) in the Norwegian EPD system (Fet et al. 2009), estimating the potential environmental impact of the chair. The functional unit (FU) of the LCA was the provision of seating for one person over 15 years. The occupational exposure in the production and disposal stages was not included, as this was outside the scope of the case study.

3.1 Measuring emission rates

The chair was submitted directly from the manufacturer to the laboratory. The chair was taken directly from the production line, wrapped twice with aluminium foil and then with non-odorous PE or PP foil and sent to the lab, shipped via overnight express. The emission rates from the chair were then measured in a test chamber with seven data points (see Table S1 in the "Electronic supplementary material" (ESM) for measured values). Chamber tests with the chair were performed according to standardised methods (ISO 2006a, b) in a 3.2-m^2 test chamber (23 °C, 50 % relative humidity and air exchange rate of 0.5/h). Samples were taken after 6, 24, 48, 72, 96, 168 and 672 h, with the chair never leaving the chamber. VOC analyses were performed according to ISO 16000–6 (ISO 2004), and volatile aldehydes C1–C6 according to ISO 16000–3 (ISO 2001). All tests and chemical analyses were performed by Eurofins AS, Galten, Denmark (Eurofins 2012).

3.2 Calculating emission model parameters

Thereafter a CNLR analysis was used to identify parameters for the three selected emission models. For the second model (Eq. (2)), three variants were included, using n values known to have a good fit with formaldehyde emissions from wood finishing (Tichenor and Guo 1991) (n=2; n=2.5; n=3). Goodness of fit of the emission models to the measured results were evaluated using R^2 (see Table S2 in the "ESM") and face validity. The model that fits the measured results best (had the highest sum of R^2) was Eq. (2) with n=2. This was chosen as the base model. This choice is further supported by the knowledge that first-order models 'almost always underestimate the long-term emissions' (Guo 2002a) and that nth-order models with n higher than 2 'may overestimate the total emissions' (Guo 2002a). The first-order and nth-order (with n=3) models were included in the calculations in order to provide an estimate of the upper and lower bounds of the emissions.

4 Results

4.1 Indoor exposure

The indoor exposure is dependent on the intake fraction, which is here determined by the parameters inhalation rate

(0.44–1.04 m^3/h), volume (150–447 m^3), air exchange (0.5–0.9/h) and mixing factor (0.1–1). The ranges are empirical values for residential homes, as presented in Hellweg et al. (2009). The variation in exposure time is 4–24 h per day. These parameters provide an estimate of the best and worst case scenarios, which may be useful for a sensitivity analysis. Based on these values the intake fraction was calculated using Eq. (4), with a range from 0.00018 to 0.1387.

Furthermore, a Nordic household scenario was defined, with values selected within common ranges. The parameters were 0.675-m^3/h inhalation rate (adult male) (Allan and Richardson 1998), 200-m^3 room volume (Øie 1998), air exchange of 0.5 (Øie 1998) and mixing factor of 0.5 (National Research Council 1991). With 8 h per day exposure time, the intake fraction for this scenario is 0.0045.

4.2 Outdoor exposure

The amount of emissions entering the environment is calculated using Eq. (6), and is dependent on the intake fraction for indoor exposure. Using the intake fraction range calculated above, the fraction of emissions entering the environment ranges from 0.8613 to 0.9998. Outdoor exposure does not require estimation of exposure time, as the USEtox multimedia fate model accounts for this. Subtracting the indoor exposure time of one person is negligible, as the outdoor population in the model is that of a continent.

4.3 Use stage impact on human health

The indoor and outdoor impact on human health from emissions in the use stage is calculated using Eqs. (7) and (8), with variations in intake fraction (low, Nordic average and high), product lifetime (1, 3, 5, 15, 30 and 70 years) and emission model (first order, n=2 and n=3). The outdoor emissions are the residual emissions when the indoor exposure has been subtracted from the total, and are modelled as emissions to urban air. Urban air has been selected instead of rural, to be on the conservative side and to provide model simplicity. Figure 2 shows impacts in a 15- and 70-year timeframe, with a product lifetime of 15 years. Figure 3 shows impacts over 70 years for three different emission models (Nordic iF). The single most significant indoor emission was found to be formaldehyde, which accounted for more than 96 % of the total impact for emission model n=2. Distinguishing between cancer and non-cancer effects revealed that cancer effects accounted for 95–99 % of the total impact for all combinations of emission model and intake fraction ($EF_{canc.}$ and $EF_{non\text{-}canc.}$ values for all substances can be found in Table S3 in the "ESM").

The results show that the largest variations in indoor impact over a 70-year period are due to variations in intake fraction (factor of 761), replacement frequency (factor of 70) and emission model (factor of 24). For 15 years, the factors for replacement frequency and emission model are lower (factor of 15 and 11, respectively). The full results can be found in Tables S4 and S5 in the "ESM".

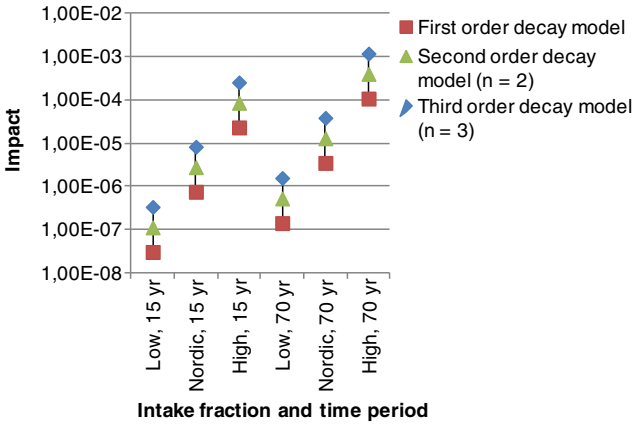

Fig. 2 Impacts in the use stage over 15 and 70 years for three emission models, with variation in intake fraction (low iF, Nordic average iF and high iF) and a furniture replacement frequency of 15 years

Variations in outdoor impact follow the same pattern as indoor impact, with the exception of intake fraction which has the opposite trend to indoor impact, and varies by a factor of 1.16. Indoor exposure is in all cases higher than outdoor exposure, with the order of magnitude varying from one to three and a half depending on intake fraction.

4.4 Life cycle assessment

A life cycle assessment of the chair (excluding the use stage) was performed using GaBi (PE International 2011) and the DATSUPI database (Fet et al. 2009), a database that has been developed specifically for the Norwegian furniture industry. Primary data were used for the furniture manufacturing processes and the most significant sub-supplier processes. Generic data were used for the background processes

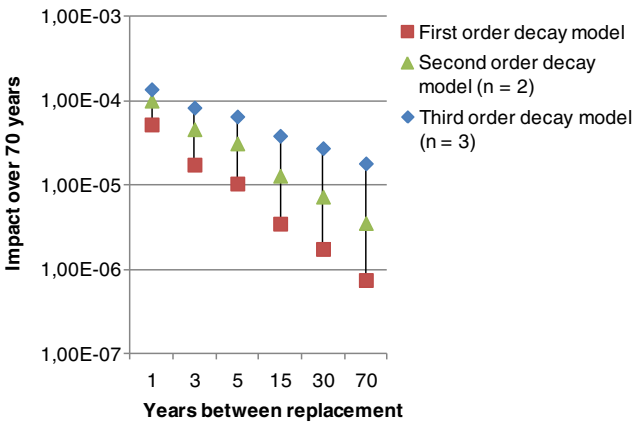

Fig. 3 Impacts in the use stage over 70 years for six different furniture replacement frequencies (three emission models, with Nordic iF)

(raw material production, energy production). For the disposal stage, generic data were used, with average Norwegian recycling and waste treatment practices for materials in the chair, as defined in DATSUPI. A high level of material and energy recovery is assumed in the disposal stage, with the environmental impacts of the recovery processes allocated to the recipient systems (as specified in the PCR). As CML impact indicators are recommended in the Norwegian EPD scheme (EPD Norway 2008), these were used here in addition to two USETox2008 categories. CML and USETox characterisation factors provided in the GaBi software (PE International 2011) were applied. The results for 11 impact categories can be found in Table S6 in the "ESM" (nine CML impact categories and two USETox2008). No normalisation or weighting methods have been applied to the potential impacts shown in this table.

The impacts on human health from the use stage were subsequently compared to the impacts from production and disposal stages. The results show that the combined indoor and outdoor impact in the use stage (Table S4 in the "ESM") varies from a factor of 6 to 4.5 orders of magnitude higher than the outdoor impact in the production and disposal stages (Table S6 in the "ESM", USETox2008 for human toxicity), taking into account that for each replacement in the use stage a new chair must also be manufactured and disposed of. The higher the intake fraction, the more important indoor exposure becomes. The choice of emission model also plays a role here. For Nordic intake fraction, the range is approximately 2 to 4 orders of magnitude between the use stage and the combined production and disposal stages, depending on emission model and replacement frequency.

5 Discussion

The discussion is structured into two parts. The first part relates to the results from the case study, including emission models, intake fraction, effect factors, replacement frequency and significance of impacts. The second part relates to relevance for stakeholders, the generic applicability of the developed method as well as areas for further research.

5.1 Case study

Uncertainty is a central element in the case study results, as the level of uncertainty of the individual elements in the model is high. When discussing uncertainty, it is important to keep in mind the uncertainty of the USEtox CFs, which is 100–1,000 (i.e. 2 to 3 orders of magnitude) (Rosenbaum et al. 2008). It should be noted that any single parameter variation is lower than the uncertainty of the USEtox model (factor of up to 761 compared to a factor up to 1,000). The uncertainty distribution and variance of the individual parameters are not known, making it difficult to construct confidence intervals.

5.1.1 Emission models

Choice of emission model influenced the case study results by a factor of 24, mainly due to differences in long-term low-level emission estimates. The three model variants included (first order, $n=2$ and $n=3$) provided values for best, worst and average scenarios. The third emission model (Eq. (3)) was not included in the case study because it had a low overall R^2 value, and because this emission model requires determining four unknown variables using CNLR analysis, with only seven data points available. Measurement uncertainty (reported by Eurofins AS to be±20 %) may also further influence the CNLR analysis. An example of this is the apparent increase in emission rate for some substances (e.g. acetone and *n*-Undecane at 72 and 96 h in Table S1 in the "ESM"), where measurement uncertainty is the most likely explanation. As emission rates can vary from substance to substance depending on the chemical properties of the individual substances (He et al. 2005), it is possible that the parameters of the emission model and of the CNLR can be refined for individual substances to provide more reliable results.

Figures 4 and 5 show calculated emission rates for the three selected emission model variants over a period of 1 month and 1 year, respectively, as well as measured emissions for the first month. The figure suggests that an *n*th-order model may exist that fits best with the measured emission rates, but determining which *n* value most accurately represents the real emission rates over time is not possible without measuring emission rates over a longer

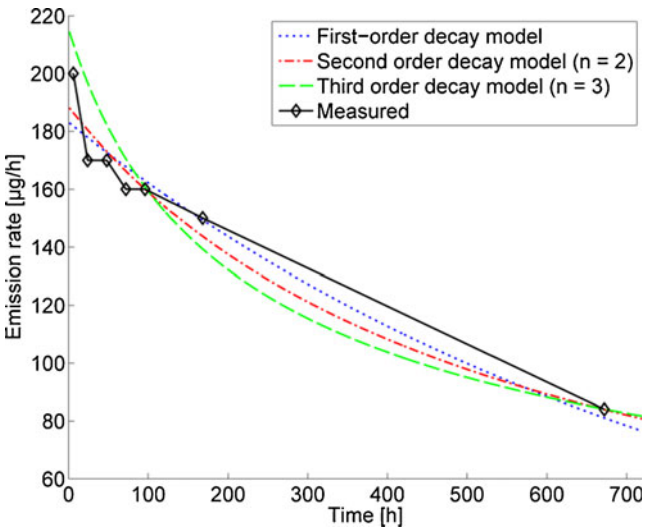

Fig. 4 Emission of formaldehyde from furniture, 1 month (micrograms per hour)

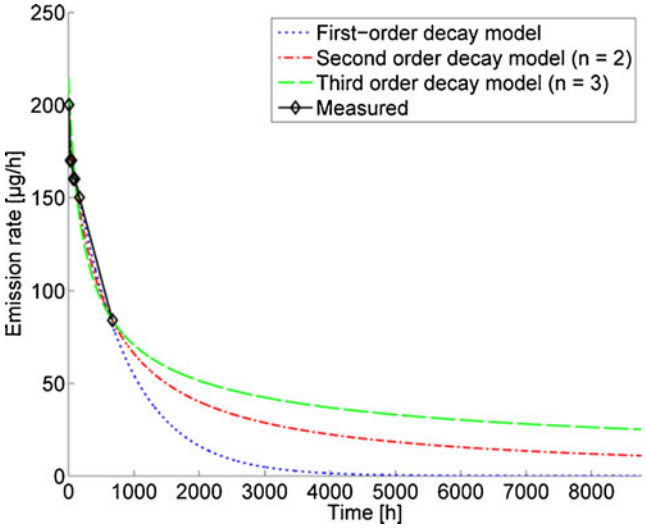

Fig. 5 Emission of formaldehyde from furniture, 1 year (micrograms per hour)

time period (months or even years). Except for the first-order emission model, all emission models evaluated here postulate that there will be continuous emissions throughout the 70-year-time period. The validity of assuming that there are emissions from the chair throughout the entire time period is unknown, as there are no experimental results to prove or disprove the assumption. Brown (1999) has shown that a dual first-order emission model can describe both short-term and long-term emission rates of formaldehyde from wood panels, but such a model was, as mentioned, excluded from the case study because of lack of data points to perform a CNLR analysis.

All emissions in the case study are assumed to be controlled by internal diffusion and vapour pressure, which were measured (although not separately). This may not be the case in real life, where factors such as sorption, desorption, humidity and temperature can influence the emission rates (Hun et al. 2010). These factors make it difficult to predict the behaviour of long-term emissions. Desorption processes in real-life situations can span over years, and are influenced by both the sink materials present in a room as well as the physical and chemical properties of the individual substances themselves (Chang et al. 1998).

5.1.2 Intake fraction

The variation in results for both 15 and 70 years showed that intake fraction was the parameter with highest variation, with a factor of 761 between best case and worst case. This is also the most difficult parameter to predict and generalise, as site specific variations will be large. The intake fraction is here dependent on size of residential home, inhalation rate, exposure time, ventilation rate and mixing factor, parameters that will vary with the age and gender of a person, the size, age and construction technique of the dwelling and from country to country. Examples of this are exposure time and inhalation rate. Inhalation rate in a Canadian study has been shown to vary from 17.54±4.06 for adult males to 12.84±2.55 for senior females (mean value±standard deviation) (Allan and Richardson 1998). Wenger et al. (2012) have proposed an intake fraction of 0.01 for volatile organic compounds. Considering the variation in the case study, this is considered to be quite close to the Nordic average intake fraction of 0.0045 defined here.

5.1.3 Effect factors

Only USEtox EFs have been used in the case study, leading to nine of the 35 emissions being assigned zero values because no data were available. Three of these nine measured emissions are aggregated totals, and have thus already been accounted for. One is a total of unspecified emissions, but as these are unknown, it is not possible to assign a meaningful EF to them. For the last five types of emissions (*n*-undecane, 2-ethylhexyl acetate, *n*-dodecane, *n*-tridecane and *n*-tetradecane) there were no USEtox CF available. In order to evaluate the potential significance of these, the emissions were compared to the lowest concentration of interest (commonly abbreviated to LCI, but not to be confused with life cycle inventory) specified in three-indoor air-quality evaluation schemes (see Table S3 in "ESM" for details). A lowest concentration of interest value is a threshold value used when evaluating or certifying building products intended to safeguard against health risk (AgBB 2010). The three schemes were the European Collaborative Action on Urban Air, Indoor Environment and Human Exposure (ECA 1997), the German Committee for Health-related Evaluation of Building Products (AgBB 2010) and the Afsset guidelines of the former French Agency for Environmental and Occupational Health Safety (Afsset 2009). This evaluation showed that USEtox covered all significant emissions in the data set.

To further evaluate the potential significance of unspecified EF*s*, an additional EF set was created. Here all unspecified EF values (marked as 'not found' or 'n/a' in Table S3 in the "ESM") were assigned the formaldehyde value in the EF_{canc} column, indented to represent a scenario where substances with unspecified EF were as harmful to human health as formaldehyde. Aggregated emissions such as TVOC were not included in this evaluation. For emissions over 15 years and using emission model $n=2$, this gave a factor of three higher impact. This is significantly lower than the uncertainty of the USEtox model itself, and can be interpreted as an indication that the most significant emissions are likely to be included in the original set of effect factors.

A limitation of the EFs used in the case study is that they only consider inhalation. Dermal contact and ingestion is

thus not included, which may lead to underestimation in particular of non-cancer effects (e.g. allergies, asthma and eczema). Other aspects that the EFs do not include are potential positive or negative interaction effects of being exposed to a cocktail of substances (Sexton et al. 2005), which can include sensitisation effects (Choi et al. 2010).

5.1.4 Replacement frequency

It has been shown that emissions of formaldehyde are persistent even in a long-term perspective in residential homes, at a level which cannot be explained by emissions from building materials alone (Hun et al. 2010; Blondel and Plaisance 2011). The case study results indicate that rapid replacement of furniture can contribute to significantly higher levels of emitted substances, with a worst case difference of a factor 70 over 70 years (first-order emission model). In the expected lifetime of the chair (15 years according to the PCR), more than 50 % of the emissions will have occurred in the first year using emission model $n=2$. With this emission model, the difference is a factor of 24. Figure 6 shows the cumulated impact over a 30-year period, with five different replacement frequencies.

The significance of replacement frequency in these findings relies on the assumption that the chair is placed in a residence from day1. If the first month of emissions occur outside of the residence, the total impact will be reduced by 14 % in a 15-year time period and 16 % in a 30-year time period (emission model $n=2$). However, with modern production techniques a chair will often be sealed in plastic at the factory and delivered directly to the consumer.

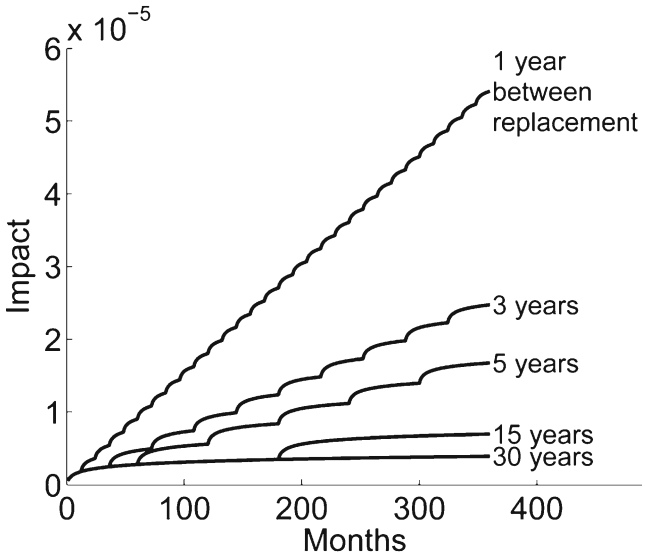

Fig. 6 Significance of furniture replacement frequency on impact in a 30-year perspective with Nordic intake fraction

5.1.5 Significance of impacts

The impacts on human health from indoor exposure in the use stage are in the case study higher than from outdoor exposure. The variation is almost entirely dependent on the intake fraction (>99 %). For best case and Nordic intake fractions in the use stage, the indoor exposures are respectively a factor of 7 and 186 higher than for outdoor exposure. It should be noted that these are not significant differences compared to the underlying uncertainty of the USEtox model. Considering how the intake fraction is defined here, a high intake fraction is likely to be due to a small room with low ventilation rate. The results show that in this case a single piece of furniture may have a high impact on the indoor air quality.

It is apparent that indoor exposure in the use stage is significantly higher (up to 6 orders of magnitude) than outdoor exposure in the production and disposal stages in most cases, except for combinations of low intake fraction with best case emission model (factor of six higher). For Nordic intake fraction, the results are 2 to 3 orders of magnitude higher. However, this result may be skewed towards the use stage for two reasons. The first is the possibility that not all hazardous emissions to outdoor is included in the production and disposal (inventory deficiency), and the second is that indoor exposure in the production and disposal stages are outside the scope of the model the case study. The significance of the latter is difficult to estimate. Here, the population is smaller than in the use stage and the majority will be healthy adults, but occupational exposure levels can also be much higher than in the use stage.

In order to assess the potential significance of the case study impacts in general, it can be useful to relate the findings to known emission or effect levels. The WHO recommendation for formaldehyde exposure is a limit of a concentration of 0.1 mg/m^3, and takes into account both cancer and non-cancer effects (WHO 2010). Using low, Nordic average and high inhalation frequencies over 70 years (4, 12 and 24 h per day) and multiplying with the USEtox EF for formaldehyde, the impact value is between 0.005 and 0.068 for a person inhaling air with maximum recommended limit over 70 years. The WHO values range from being almost identical to the case study results (indoor and outdoor in the use stage) to 6 orders of magnitude higher. With Nordic average intake fraction the WHO values are 1 to 4 orders of magnitudes higher (whereas the uncertainty within the USEtox model is 2 to 3 orders of magnitude). Although the WHO limit values are risk based and thus not directly comparable to the results of the case study, the results are interpreted as an indication that there are indoor exposure scenarios where a single piece of furniture can be of significance.

5.2 Applicability of the developed method

The results of the case study show that manufacturers that intend to investigate the impact on human health from toxic substances in the value chain, should include consumers. Neglecting to do so means that potentially significant health impacts are overlooked, which can lead to problem shifting from the production to the use stage. Any investigation should include long-term as well as short-term effects. The results here support previous recommendations to include human health impacts from indoor emissions into LCA (Hellweg et al. 2009; Wenger et al. 2012).

5.2.1 Relevance for stakeholders

Two stakeholder groups can be identified that are directly affected by the emissions from products (in addition to the general population that is affected through increased background levels of exposure). The first group is consumers and the second is workers in the production and disposal stages.

Methods for integrating occupational health aspects in an LCA have been developed (Kim and Hur 2009; Andrews et al. 2009), but usually without including the consumer in the use stage. For consumers, the case study results show that with high replacement frequency and poor ventilation the impact on human health from indoor exposure from the chair can be significant, when compared to exposure at the recommended WHO formaldehyde exposure limit. However, the impact may be higher or lower for other product groups (e.g. other types of furniture, toys, carpets or building materials) or when looking at multiple products in a room. Knowing that background levels of formaldehyde concentrations in residential homes tend to stabilise at a relatively high level that is not alleviated by increased ventilation, removing sources is a recommended approach (Hun et al. 2010). Other approaches to reducing indoor exposure from products where the emissions have an exponential decay rate, are increasing ventilation rates in the first months after a purchase, 'airing out' the product before using it (e.g. in a storage room for a period of a month) or acquiring second hand products instead of buying new.

Stakeholders in the production and disposal stage can be distinguished into two groups, where a person may belong to one or both: exposed workers and people at a position to influence emission rates and exposure (e.g. designers, purchasers and occupational hygienists). Emissions from the product itself in the disposal stage are arguably less relevant, in part because of the low level of long-term emissions and in part because of potential exposure to other emissions arising from disposal processes are more significant (e.g. hexavalent chromium arising from leather incineration (Chen et al. 1998)).

5.2.2 Recommendations and limitations

The method presented through Eqs. 1, 2, 3, 4, 5, 6, 7 and 8 proposes to calculate the impact on human health from emissions in the use stage, based on a set of measured emission rates. The case study has shown that the model can be applied to chairs, and that the use stage can be significant when compared to outdoor exposure in the production and disposal stages. The method is not specific to furniture, it is intended to be generic and applicable to any type of product where the emission rate can be modelled using an exponential decay model (i.e. Eqs. (1), (2) and (3)). The model is based on the assumption that there is an *n*th-order decay model that can describe the emission rate over time, where the optimal *n* value can vary from one product type to another. For products that are similar to a chair, the method can be applied in a similar manner as in the case study. This applies to products that have long product lifetime and with only one person exposed per product. For other types of products, the following should be considered:

- If the product lifetime is short, it is likely that a first-order emission model (Eq. (1)) can perform as well as an *n*th-order emission model (with $n>1$) (Eq. (2)), thus simplifying the emission model.
- If multiple persons are exposed, this must be accounted for when calculating the intake fraction (Eq. (4)).

In all cases the amount of product(s) (e.g. in units, kilogramme or square metre) should be related to a meaningful functional unit. Furthermore, the results of the case study show that intake fraction and replacement frequency are the most important model parameters. These are also the two most difficult to generalise, as they are entirely dependent on the characteristics of the user and the user's residence. Recognising the level of uncertainty associated with these two parameters, it is recommended that when a sensitivity analysis is performed, the variation in these parameters is always included (for example by using best and worst case estimates).

When applying the method developed here, there are three limitations that the user should be aware of. The first is that only the emissions from the studied product(s) are included, underlining the importance of defining a functional unit that is useful for the intended audience of the LCA. The second is the challenge of defining a relevant intake fraction. This is especially relevant if the purpose of the LCA is to evaluate the relative importance of life cycle stages (i.e. hot spot analysis), but of less importance when performing comparative studies between product systems. The third is that the one-box model used to model indoor exposure does not take into consideration sink effects in the room where the product is located, thus potentially underestimating the long-term emission rates (Chang et al. 1998).

Furthermore, performing a CNLR analysis to find parameters for the emission model requires that there are enough data points. This means that it is not possible to find emission model parameters based on a single emission rate only measured at 28 days, which is required for some types of certification (Nordic Ecolabelling 2011a, b). It is therefore recommended to perform additional measurements during the 28-day interval the product is in the test chamber.

5.2.3 Further research

A number of challenges remain to make the suggested method accessible as a general method that can be used broadly. The first challenge is, as already mentioned, the lack of knowledge on long-term emissions. The measurements performed here ran for less than a month, whereas the lifetime of many products span over decades. The second challenge is also related to emission rates. Laboratory measurements are accurate, but also costly, time consuming and usually do not include long-term emission rates. Developing a calculation methodology that can accurately predict short-term and long-term emissions based on product design and material content will make it feasible to include these aspects already in the design phase, for example integrated in CAD software. Such models can also have societal relevance, considering the scale of emissions from products in a national or regional perspective (Rydberg et al. 2012). The third challenge is the relationship between the calculated intake fraction and real-life situations, where factors such as mixing, sorption, desorption and multiple emission sources and sinks are present. In general, the effect of adsorption and desorption on material surfaces influence the indoor air during the entire time the product/furniture is in a building (Berglund et al. 1989). Improvements in these areas can reduce the level of uncertainty in the results and improve the precision.

6 Conclusions

This paper has presented a methodology, based on experimentally measured emission rates, which can be used to estimate a product's impact on human health in the use stage when performing an LCA. Three emission models (first order, *n*th-order and dual first order) have been evaluated, identifying an *n*th-order model as best suited when there are few data points and long-term emissions are of concern. The results of a case study of a chair have shown that the impact on human health from indoor emissions can be significant, when compared with the impact from outdoor emissions in the other life cycle stages. These impacts are therefore recommended to be considered when performing an LCA of products that emit volatile organic compounds (VOCs). The method is applicable to all products that emit VOCs, provided that the emission rate can be modelled using an exponential decay model and that the product amount is related to a meaningful functional unit. It is recognised that without specific exposure data (e.g. ventilation rates, sorption) the uncertainty will be high. In this case, the developed method can be used to indicate if the use stage is a hot spot that should be further investigated. Additional research on long-term emissions in real-life situations is needed to reduce the level of uncertainty.

The case study has shown that under certain circumstances, the impact from exposure in the use stage may alone be high enough to raise concern. Recommended actions to reduce use stage exposure are limiting early exposure (more than 14 % of emissions may occur in the first month and more than 50 % in the first year) and replacing furniture less frequently. The results favour re-use of furniture, as this will eliminate early exposure for the next users.

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References

Afsset (2009) Saisine Afsset no. 2004/11. Afsset, Paris

AgBB (2010) Evaluation procedure for VOC emissions from building products. http://www.agbb-nik.de/. Accessed 20 September 2012

Allan M, Richardson GM (1998) Probability density functions describing 24-hour inhalation rates for use in human health risk assessments. Hum Ecol Risk Assess 4(2):379–408

Andrews E, Lesage P, Benoît C, Parent J, Norris G, Revéret J-P (2009) Life cycle attribute assessment—case study of Quebec greenhouse tomatoes. J Ind Ecol 13(4):565–578

Bennett DH, McKone TE, Evans JS, Nazaroff WW, Margni MD, Jolliet O, Smith KR (2002) Defining intake fraction. Environ Sci Technol 36(9):206a–211a

Berglund B, Johansson I, Lindvall T (1989) Volatile organic compounds from used building materials in a simulated chamber study. Environ Int 15:383–387

Blondel A, Plaisance H (2011) Screening of formaldehyde indoor sources and quantification of their emission using a passive sampler. Build Environ 46(6):1284–1291

Brasche S, Bischof W (2005) Daily time spent indoors in German homes—baseline data for the assessment of indoor exposure of German occupants. Int J Hyg Environ Heal 208(4):247–253

Brown SK (1999) Chamber assessment of formaldehyde and VOC emissions from wood-based panels. Indoor Air 9(3):209–215

Chang JCS, Sparks LE, Guo Z, Fortmann R (1998) Evaluation of sink effects on VOCs from a latex paint. J Air Waste Manag Assoc 48 (10):953–958

Chen J-C, Wey M-Y, Chiang B-C, Hsieh S-M (1998) The simulation of hexavalent chromium formation under various incineration conditions. Chemosphere 36(7):1553–1564

Choi H, Schmidbauer N, Sundell J, Hasselgren M, Spenger J, Bornehag C-G (2010) Common houshold chemicals and the allergy risks in pre-school children. PLoS One 5(10):e13423

Colombo A, De Bortoli M, Knöppel H, Pecchio E, Visser H (1993) Adsorption on selected volatile organic compounds on a carpet, a

wall coating, and a gypsum board in a test chamber. Indoor Air 3:276–282
ECA (1997) Evaluation of VOC emissions from building products. European Comission, Brussels
EPD Norway (2008) Product category rules for preparing an Environmental Product Declaration (EPD) for product group seating solution. EPD Norway, Oslo
Eurofins (2012) Eurofins. http://www.eurofins.com. Accessed 20 September 2012
Fet AM, Skaar C, Michelsen O (2009) Product category rules and environmental product declarations as tools to promote sustainable products: experiences from a case study of furniture production. Clean Technol Environ 11(2):201–207
Greenguard (2011) GREENGUARD Environmental Institute Formaldehyde Free Verification Requirement. Marietta
Guo Z (2002a) Review of indoor emission source models. Part 1. Overview. Environ Pollut 120(3):533–549
Guo Z (2002b) Review of indoor emission source models. Part 2. Parameter estimation. Environ Pollut 120(3):551–564
He G, Yang X, Shaw CY (2005) Material emission parameters obtained through regression. Indoor Built Environ 14(1):59–68
Hellweg S, Demou E, Bruzzi R, Meijer A, Rosenbaum RK, Huijbregts MAJ, McKone TE (2009) Integrating human indoor air pollutant exposure within life cycle impact assessment. Environ Sci Technol 43(6):1670–1679
Hun DE, Corsi RL, Morandi MT, Siegel JA (2010) Formaldehyde in residences: long-term indoor concentrations and influencing factors. Indoor Air 20(3):196–203
International Organization for Standardization (2001) Indoor air—part 3: determination of formaldehyde and other carbonyl compounds - Active sampling method (ISO 16000–3:2001). International Organization for Standardization, Geneva
International Organization for Standardization (2004) Indoor air—part 6: determination of volatile organic compounds in indoor and test chamber air by active sampling on Tenax TA sorbent, thermal desorption and gas chromatography using MS/FID (ISO 16000–6:2004). International Organization for Standardization, Geneva
International Organization for Standardization (2006a) Environmental management—life cycle assessment—requirements and guidelines (ISO 14044:2006). International Organization for Standardization, Geneva
International Organization for Standardization (2006b) Indoor air—part 9: determination of the emission of volatile organic compounds from building products and furnishing—emission test chamber method (ISO 16000–9:2006). International Organization for Standardization, Geneva
International Organization for Standardization (2006c) Indoor air—part 11: determination of the emission of volatile organic compounds from building products and furnishing—sampling, storage of samples and preparation of test specimens (ISO 16000–11:2006). International Organization for Standardization, Geneva
Jørgensen RB, Bjørseth O (1999) Sorption behaviour of volatile organic compounds on material surfaces—the influence of combinations of compounds and materials compared to sorption of single compounds on single materials. Environ Int 25:17–27
Jørgensen RB, Bjørseth O, Malvik B (1999) Chamber testing of adsorption of volatile organic compounds (VOCs) on material surfaces. Indoor Air 9(1):2–9
Jørgensen RB, Dokka TH, Bjørseth O (2000) Introduction of a sink–diffusion model to describe the interaction between VOCs and material surfaces. Indoor Air 10:27–38
Kim I, Hur T (2009) Integration of working environment into life cycle assessment framework. Int J Life Cycle Assess 14(4):290–301
Kuehn BM (2008) Stronger formaldehyde regulation sought. JAMA 299(17):2015–2015
Meijer A, Huijbregts MAJ, Reijnders L (2005) Human health damages due to indoor sources of organic compounds and radioactivity in life cycle impact assessment of dwellings—part 1: characterisation factors. Int J Life Cycle Assess 10(5):309–316
National Research Council (1991) Human exposure assessment for airborne pollutants: advances and opportunities. National Academy Press, Washington, DC
Nordic Ecolabelling (2011a) Nordic Ecolabelling of furniture and fitments. Nordic Ecolabelling, Oslo
Nordic Ecolabelling (2011b) Nordic Ecolabelling of panels for the building, decoration and furniture industries. Nordic Ecolabelling, Oslo
Øie L (1998) The ventilation rate of 344 Oslo residences. Indoor Air 8 (3):190–196
PE International (2011) GaBi. http://www.gabi-software.com. Accessed 20 September 2012
Rosenbaum R, Bachmann T, Gold L, Huijbregts M, Jolliet O, Juraske R, Koehler A, Larsen H, MacLeod M, Margni M, McKone T, Payet J, Schuhmacher M, van de Meent D, Hauschild M (2008) USEtox—the UNEP-SETAC toxicity model: recommended characterisation factors for human toxicity and freshwater ecotoxicity in life cycle impact assessment. Int J Life Cycle Assess 13 (7):532–546
Rydberg T, Westerdahl J, Hallberg E, Öman A, Andersson PL, Haglund P, Holmgren T, Fuhrman F, Molander S, Tivander J (2012) Emissions of additives from plastics in the societal material stock: a case study for Sweden. In: Bilitewski B, Darbra RM, Barceló D (eds) The handbook of environmental chemistry volume 18. Springer, Berlin, pp 133–149
Saarela K, Tirkkonen T (2004) M1, emission classification of building materials: protocol for chemical and sensory testing of building materials. VTT, Finland
Salthammer T, Mentese S, Marutzky R (2010) Formaldehyde in the indoor environment. Chem Rev 110(4):2536–2572
Sexton K, Adgate JL, Church TR, Ashley DL, Needham LL, Ramachandran G, Fredrickson AL, Ryan AD (2005) Children's exposure to volatile organic compounds as determined by longitudinal measurements in blood. Environ Health Perspect 113 (3):342–349
Sexton K, Adgate JL, Ramachandran G, Pratt GC, Mongin SJ, Stock TH, Morandi MT (2003) Comparison of personal, indoor, and outdoor exposures to hazardous air pollutants in three urban communities. Environ Sci Technol 38(2):423–430
Singer BC, Hodgson AT, Hotchi T, Ming KY, Sextro RG, Wood EE, Brown NJ (2007) Sorption of organic gases in residential rooms. Atmos Environ 41(15):3251–3265
SPSS (2010) SPSS 14.0 for Windows
Tichenor BA, Guo Z (1991) The effect of ventilation on emission rates of wood finishing materials. Environ Int 17(4):317–323
Tichenor BA, Guo Z, Dorsey JA (1992) Emission rates of mercury from latex paints. U.S Environmental Protection Agency, Washington DC
Tichenor BA, Guo Z, Dunn JE, Sparkds LE, Mason MA (1991) The interaction of vapour phase organic compounds with indoor sinks. Indoor Air 1:23–37
USEtox (2011) USEtox™ model. http://www.usetox.org. Accessed 8 February 2011
Wenger Y, Li D, Jolliet O (2012) Indoor intake fraction considering surface sorption of air organic compounds for life cycle assessment. Int J Life Cycle Assess 17(7):919–931
WHO (2010) WHO guidelines for indoor air quality: selected pollutants. The WHO European Centre for Environment and Health, Bonn